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ORIGINAL ARTICLE

Nutrient export by rivers to the coastal waters of China: management strategies and future trends

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Abstract We analyzed past and future trends in river export of dissolved nitrogen (N) and phosphorus (P) to the coastal waters of China, for a selection of rivers, as calculated by the Global *NEWS* models (Nutrient Export from WaterSheds). Over the period 1970–2000, river export of dissolved nutrients increased considerably, especially of dissolved inorganic N and P. Anthropogenic sources of N and P in rivers have become increasingly important, in particular N and P losses from agriculture. We tested the sensitivity of calculated nutrient export by rivers to changes in selected controlling factors. The calculated export of dissolved N and P is sensitive to changes in hydrology, synthetic fertilizer use and manure excretion in river basins. Changes in sewage inputs have large effects on dissolved inorganic P export. We analyzed future trends up to 2050 for several scenarios to illustrate the combined effects of selected management options. In the Base Scenario, the N and P export by the selected rivers increases considerably between 2000 and 2030. The increase continues from 2030 to 2050 except for DIP. Strategies to

reduce N and P export by rivers include changes in agriculture, sewage and energy use. Changes in agriculture have the largest impact on future dissolved N and P river export to the coastal waters of China. The effectiveness of reduction strategies differs between nutrient forms and basins.

Keywords Nutrient export · River · Coastal waters · Management strategy · Sensitivity analysis · Scenario

Introduction

Events of coastal eutrophication and hypoxia have been reported in many world regions, including the coastal waters of China (Diaz and Rosenberg 2008; Selman et al. 2008). Increasing nutrient export from land to sea has been reported as the most important causes of China's coastal water pollution. China has experienced 30 years of rapid economic growth, and the population is currently the largest of the world. The increasing human activities in China's river basins pose a long-term threat to the quality of the coastal waters (Qu and Kroeze 2010). China's largest River, the Changjiang (Yangtze River), is a typical example. The Changjiang basin covers about 20% of the total area of China and 35% of the country's population (Xing and Zhu 2002). The nutrient export by this river has been increasing fast. For instance, the annual river nitrate–N flux increased from 0.1 Tg in the year 1968 to 1.2 Tg in the year 1997 (Yan et al. 2003). This is caused by the increased use of synthetic fertilizer and discharge of industrial wastewater and domestic sewage (Gao and Wang 2008). The increasing inputs of nutrients have affected the coastal water quality and ecosystems. In the East China Sea,

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harmful algal blooms occurred only several times in the 1970s, but 86 times in 2003 (UNEP 2007; Jin et al. 1996).

The coastal waters of China deserve particular attention not only because of the serious current conditions, but also because of potential future trends. The Chinese population and economy are projected to continue to grow fast. Between 2000 and 2020, China's population is expected to increase from 1.27 billion to 1.45 billion, and the GDP per capita from \$856 person⁻¹ year⁻¹ to \$4,800 person⁻¹ year⁻¹ (in constant US dollars of 2000) (Jiang et al. 2007; Louis 2009). As a consequence, the anthropogenic nutrient inputs to watersheds are expected to increase. This holds for N and P inputs from synthetic fertilizers and animal manure application in agriculture, but also for sewage and industrial inputs. Climate change may add to this if global warming is associated with increased runoff and river discharge (Qu and Kroeze 2010).

To reduce coastal water pollution in China, national mitigation strategies as well as regional action plans are needed. To achieve this, it is important to understand the mechanisms controlling both nutrient sources and nutrient removal in watersheds and river networks. This holds in particular for the understanding of the magnitude and geographic distribution of nutrient export by rivers to the coastal waters of China, and the sources of these nutrients. Such understanding may help us to identify effective options for the management of nutrients, land use and water resources, in order to reduce nutrient export by Chinese rivers in the future. The purpose of this study is, therefore, to explore the future trends in nutrient export by Chinese rivers and the potential management strategies. To achieve this, an integrated modeling approach is needed.

Quantitative analyses of coastal eutrophication and policy options to reduce this pollution are scarce in China. Lack of systematic measurements of water quality is an important reason for this (Li et al. 2007). Also, there is a lack of systematic analyses of nutrient budgets at the pan-Chinese scale. Many studies estimate nutrient export as the product of nutrient concentrations and mean runoff, without analyzing human influences (Yan et al. 2010). In this study, we use the spatially explicit Global *NEWS* (Global Nutrient Export from WaterSheds) model, which models nutrient export by world rivers to coastal seas, as affected by human activities on land. The early versions of the *NEWS* models (*NEWS*-1) (Seitzinger et al. 2005; Dumont et al. 2005; Beusen et al. 2005; Harrison et al. 2005a, b) and the recently improved versions (*NEWS*-2) (Fekete et al. 2010; Bouwman et al. 2009; Van Drecht et al. 2009; Mayorga et al. 2010; Seitzinger et al. 2010) have been developed to analyze past and future trends in nutrient export by rivers.

The present paper follows up on the work of Qu and Kroeze (2010). They applied the *NEWS*-2 models to

analyze river export of nutrients to the coastal waters of China for the period 1970–2050 for the Millennium Ecosystem Assessment (MEA) scenarios. They presented and interpreted the *NEWS* model results for the largest rivers of China. A comparison of model results with measurements, indicating that the *NEWS* models can reasonably describe the N and P export by Chinese rivers. They analyzed the Chinese continental contribution to global nutrient export, as well as past and future trends in driving forces and nutrient export (magnitude, composition and sources) for the MEA scenarios.

In this paper, we focus on options to reduce N and P export by selected rivers to the coastal waters of China on the basis of *NEWS* model calculations. First, we apply the *NEWS* models to identify the important sources of nutrients in Chinese rivers through an analysis of past trends (1970–2000). Next, we test the sensitivity of calculated nutrient exports by rivers to changes in selected controlling factors. Third, we develop scenarios illustrating the combined effects of selected management options. For the nutrient forms, we focus on dissolved forms of N and P, including dissolved inorganic N and P (DIN and DIP) and dissolved organic N and P (DON and DOP). Even though the Global *NEWS* models also include particulate forms of N and P, we do not consider these here for three reasons. First, the *NEWS* models perform less well for particulate N and P than for dissolved forms for the selected Chinese rivers (Qu and Kroeze 2010). Second, dissolved forms are generally considered the most important pollutants in coastal waters of China (SOA 2007). Third, river loadings of particulate N and P have been decreasing in recent years. This is because sediment export by Chinese rivers decreased significantly, due to reduced precipitation, increased human water use, increased sediment retention behind dams and in reservoirs, and improved sediment control (Qu and Kroeze 2010; Syvitski et al. 2005).

Method

Study area

Our study area includes 13 river basins draining into Chinese seas (Fig. 1; Table 1). We present *NEWS*-2 model runs. *NEWS*-2 model runs are based on hydrology from the Water Balance Model (WBMplus), including over 6000 river basins as defined in STN-30p (version 6.01, 0.5° × 0.5° resolution) (Vorosmarty et al. 2000a, b). In the STN-30p system, there are more than 90 exoreic river basins draining into the coastal waters of China. From these river basins, we selected the 13 largest (>25,000 km²) since the *NEWS* model results are relatively uncertain for smaller basins (Vorosmarty et al. 2000b; Harrison et al.

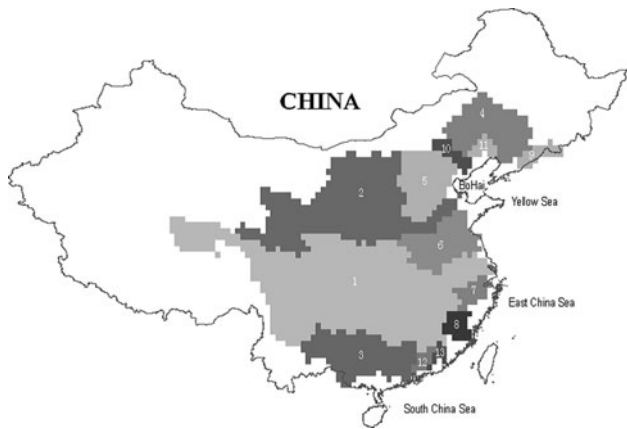


Fig. 1 The selected 13 exoreic river basins. See Table 1 for details

2005b). The 13 selected river basins cover an area of 4.2 million km², which is about 90% of the area of basins draining into the coastal waters of China. The basin areas range from about 25,000 km² to 1.8 million km². Among the 13 are three large basins: the Changjiang (Yangtze River), the Huanghe (Yellow River) and the Zhujiang (Pearl River), covering a considerable part of the Chinese continent. In the year 2000, about 1 billion people lived in these 13 river basins, of which two-thirds in rural areas. The Haihe, Huaihe and Fuchunjiang basins have the highest population densities and are most urbanized, while the Luanhe and Dalinghe basins have the lowest urban population densities. The hydrological conditions differ significantly among rivers. The river basins in southern China, such as Zhujiang, Fuchunjiang, Menjiang, Dongjiang and Hanjiang, have a relatively high annual precipitation (mm year⁻¹) and runoff (m year⁻¹). Basins located

in central and northern China, including Huanghe, Liaohe, Haihe, Luanhe and Dalinghe are generally drier.

NEWS model

The global *NEWS* models are spatially explicit models for predicting basin scale export of dissolved inorganic nitrogen and phosphorus (DIN, DIP), dissolved organic carbon, nitrogen and phosphorus (DOC, DON, DOP), total suspended solids (TSS), particulate organic carbon (POC), particulate nitrogen and phosphorus (PN and PP) and dissolved silica (DSi) (Seitzinger et al. 2005).

The recently improved version of *NEWS* models (*NEWS-2*) enhances the methods for estimating some sources of nitrogen inputs, updates the model inputs (the base year was updated from 1995 to 2000) and develops a unified modeling environment to facilitate the analyses of river nutrient exports of different scenarios and comparisons by basin and by element or form (Mayorga et al. 2010). *NEWS-2* includes a unified sub-model of dissolved nutrient forms (hereafter, we refer it as *NEWS-dissolved* model) and a sub-model of particulates. The dissolved sub-model is broadly based on a mass-balance approach for the land surface (watershed) and river system, which enables analyses of individual source contributions to total export at the basin mouth. The particulates sub-model is largely statistical and can therefore not be used for a detailed source attribution (Mayorga et al. 2010; Qu and Kroeze 2010; Seitzinger et al. 2010). This study focuses on river dissolved N and P export, which were analyzed by the *NEWS-dissolved* model. The *NEWS-2* modeled loads of dissolved N and P for the selected 13 Chinese

Table 1 Overview of the selected 13 rivers draining into the coastal waters of China (modified from Qu and Kroeze (2010))

Basin ID	Basin name	Area (10 ³ km ²) ^b	Latitude band ^b	Sea basin ^b
1	Changjiang	1,792	23.5–35N	East China Sea
2	Huanghe	893	35–50N	Bohai Gulf
3	Zhujiang	408	10–23.5N	North South China Sea
4	Liaohe	274	35–50N	Bohai Gulf
5	Haihe	245	35–50N	Bohai Gulf
6	Huaihe	244	23.5–35N	West Yellow Sea
7	Fuchunjiang	67	23.5–35N	East China Sea
8	Menjiang	66	23.5–35N	East China Sea
9	Yalu ^a	51	35–50N	East Yellow Sea
10	Luanhe	46	35–50N	Bohai Gulf
11	Dalinghe	32	35–50N	Bohai Gulf
12	Dongjiang	34	10–23.5N	North South China Sea
13	Hanjiang	25	23.5–35N	North South China Sea

^a Yalu is a river at the border of China and North Korea and eventually drains into the Yellow Sea between Dandong (China) and Sinŭiju (North Korea)

^b Source *NEWS-2* (Mayorga et al. 2010); Basins are defined in STN-30p (version 6.01, 0.5° × 0.5° resolution) (Vorosmarty et al. 2000a, b)

river basins compare well to measured loads (Qu and Kroeze 2010).

Figure 2 shows the unified framework of *NEWS*-dissolved model, modified from Mayorga et al. (2010). The dissolved model incorporates inputs to rivers from both diffuse sources and point sources. The diffuse sources inputs to watersheds include both natural and anthropogenic sources. Some of the diffuse inputs are calculated explicitly, including synthetic fertilizers, manure, N fixation and atmosphere N deposition, as a function of specific crop types, land use, animal type, etc. Other diffuse inputs are calculated based on export coefficient approach, including P weathering from phosphorus-containing minerals, dissolved organic N and P leaching from land to streams (prior to synthetic fertilizer and manure inputs). The watershed nutrient removals are expressed as functions of water runoff. The export of element E as dissolved form F originating in explicit diffuse sources ($RS_{dif_{expl,F}}$) is estimated using a watershed export fraction term ($FE_{ws,F}$; 0–1), which is calculated by a calibrated, unitless watershed export constant and a runoff modulation ($f_F(R_{nat})$). The DIP, DON and DOP export to rivers from export coefficient diffuse sources ($RS_{dif_{ec,F}}$) are modulated by runoff using the same expressed runoff function $f_F(R_{nat})$. The point-source emissions of N and P from watershed ($WShw_E$) into rivers include sewage from human excrement and detergents from household and industrial activities, which are estimated from national and regional socioeconomic and sanitation information (Van Drecht et al. 2009). The diffuse and point sources inputs to rivers may experience retention process in rivers resulted by both natural and anthropogenic factors, including denitrification, removal in large reservoirs and through consumptive water abstractions (estimated from total river discharge at the river mouth before and after the implementation of irrigation, domestic use, industrial use and other water withdrawal schemes).

Mayorga et al. (2010) describe how the dissolved N and P sub-models in *NEWS*-2 were recalibrated, using largely the same procedures in *NEWS*-1 (Dumont et al. 2005; Harrison et al. 2005a, b), newly estimated nutrient inputs and observed hydrological factors for the year 2000, and a dataset of observed annual nutrients export in river mouths. The Nash–Sutcliffe model efficiency (R^2) on log-transformed basin yields was used in model calibration and validation (Mayorga et al. 2010). The *NEWS*-2 dissolved models can explain 50–90% of the observed variation in nutrients export by world rivers (Mayorga et al. 2010). The models results for the selected Chinese rivers compare well with observed data (Qu and Kroeze 2010). For details of *NEWS* models, we refer to the original model descriptions (Seitzinger et al. 2005; Dumont et al. 2005; Beusen et al. 2005; Harrison et al. 2005a, b) and *NEWS*-2 descriptions

(Mayorga et al. 2010; Fekete et al. 2010; Bouwman et al. 2009; Van Drecht et al. 2009; Seitzinger et al. 2010).

Sensitivity analysis

We first analyzed past nutrient export to the coastal waters of China by the selected 13 rivers using historical input data for 1970 and 2000. From the analyses of past trends, we identified factors that influence nutrient export to the coastal waters of China. Next, we analyzed the sensitivity of the calculated nutrient export by rivers to changes in these identified factors. We focused on changes in model parameters that would result in reduced nutrient export to coastal waters. To this end, several important model inputs were changed (increased or decreased) by an arbitrary 30%. Factors influencing nutrient export include both the nutrient inputs to watersheds/rivers (including synthetic fertilizers, manure, atmospheric N deposition, and sewage from human excrement and detergents) and the physical and hydrological process within the watersheds and river networks (including runoff, nutrient retention in reservoirs, consumptive water use and nutrient removal by sewage treatment). All sensitivity analyses were done for the year 2000. We analyzed the mean changes in nutrients export in the selected 13 basins, and the ranges in reductions in nutrient export in individual basins.

Scenarios analysis

We analyzed future trends in nutrient export for the years 2030 and 2050 for five scenarios: a Base Scenario (BS) and four Reduction Scenarios (RSs).

The Base Scenario (BS) is the Global Orchestration scenario of the Millennium Ecosystem Assessment (MEA) (Fekete et al. 2010; Alcamo et al. 2005; Bouwman et al. 2009; Van Drecht et al. 2009; Qu and Kroeze 2010). In the Base Scenario, globalization is the trend, and economic development the major focus. Moreover, the scenario assumes a reactive approach toward ecosystem management, implying that pollution problems are not tackled before they become apparent. In this scenario, the population in the selected basins will continue to grow. The economy develops fast, resulting in a large demand for food, especially for meat and milk. Synthetic fertilizer use and animal waste production are assumed to continue to increase fast, in line with population and dietary trends. The scenario also assumes improved sanitation and sewage connection in the selected basins. As a result, more sewage from human waste and detergents are discharged into rivers. Greenhouse gas emissions are assumed to increase rapidly, resulting in climate change and thus influencing river discharge and runoff. Hydropower of China is assumed to increase significantly, especially considering

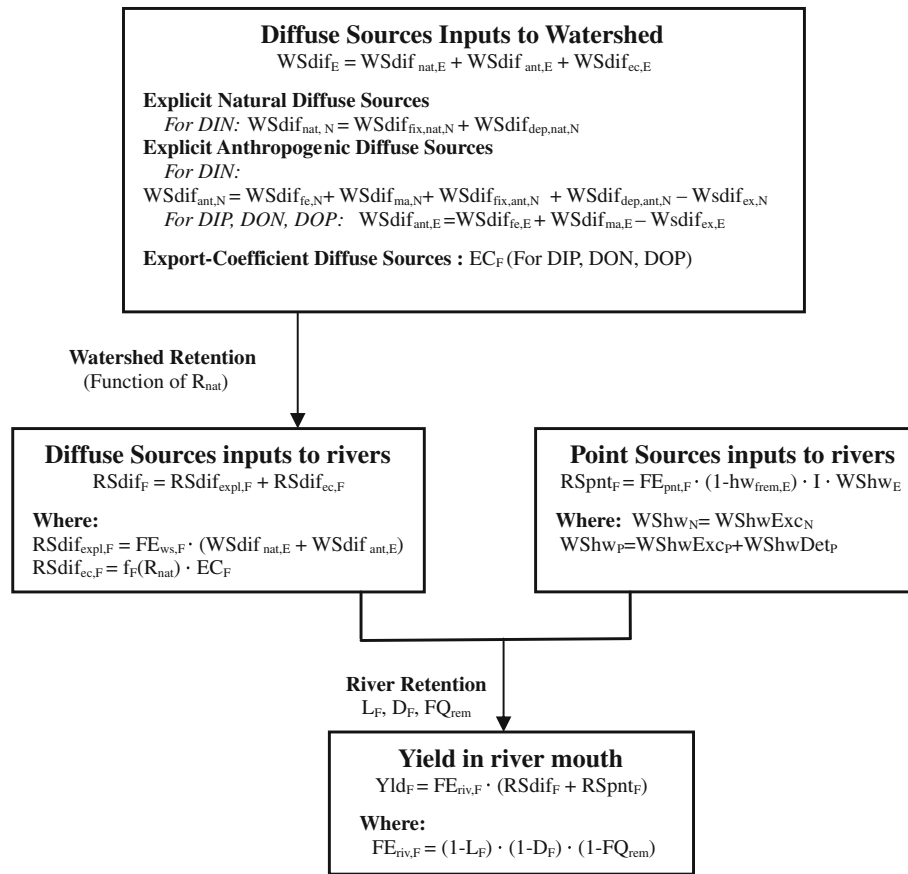


Fig. 2 An overview of the dissolved nutrient model of NEWS-2 (Mayorga et al. 2010). Note: D_F —reservoir retention factor for nutrient form F for basin (D_{din} and D_{dip}; for dissolved organic matter, $D_F = 0$) (0–1); **E (subscript)**—nutrient element (N and P); EC_F —A globally constant, calibrated export coefficient term that is identical over natural and agricultural areas, for DIP, DON and DOP. For DIP, represents weathering of phosphorus-containing minerals, while for DON and DOP it corresponds to a net leaching or export of dissolved organic matter from land into streams ($\text{kg km}^{-2} \text{ year}^{-1}$); **F (subscript)**—nutrient form (DIN, DIP, DON and DOP); $FE_{pnt,F}$ —fraction of total point-source export of element E emitted as form F, $FE_{pnt,DIN} = 0.485 + 0.255 \cdot [hw_{freem,N}/\max(hw_{freem,N})]$, for other N and P forms it is equal to a calibrated or default constant, c_F (0–1); $FE_{riv,F}$ —fraction of F inputs to rivers (point + diffuse) that is exported at the basin mouth, corresponding to retention within the river system ($1 - FE_{riv,F}$) (0–1); $FE_{ws,F}$ —fraction of F inputs to watershed exported to rivers, a runoff modulation function with a calibrated unitless watershed export constant (0–1); $f_F(R_{nat})$ —a runoff modulation function with one or two globally calibrated coefficients and with a shape determined empirically for each nutrient form; FQ_{rem} —fraction of discharge removed by consumptive water use (identical for all forms), calculated by river discharge under natural, climate-driven conditions (Q_{nat}) and river discharge under actual, anthropogenic conditions (Q_{act}) (0–1); $hw_{freem,E}$ —fraction of element E in sewage influent removed via wastewater treatment (0–1); **I**—fraction of population connected to a sewage system (0–1); L_F —retention fraction along the river network (channel, river bed sediments, hyporheic areas and floodplains); $L_F = 0$ for all other forms except DIN (0–1); R_{nat} —mean annual runoff (m year^{-1}) from land to streams, before consumptive water abstractions through water engineering (m year^{-1}); **RS**—river sources; $RSdif_{ec,F}$ —F input to

rivers from diffuse sources estimated via export coefficients ($\text{kg km}^{-2} \text{ year}^{-1}$); $RSdif_{expl,F}$ —F input to rivers from explicit diffuse sources ($\text{kg km}^{-2} \text{ year}^{-1}$); $RSdif_F$ —the export of F from the watershed to streams via diffuse sources, both natural and anthropogenic ($\text{kg km}^{-2} \text{ year}^{-1}$); $RSpt_F$ —the export of F from the watershed to streams via point sources (sewage) ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{ant,E}$ —net E input to watershed (land) from diffuse sources, on agricultural areas ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{dep,ant,N}$ —N input to watershed from atmospheric N deposition on agricultural land ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{dep,nat,N}$ —N input to watershed from atmospheric N deposition on natural areas ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_E$ —diffuse-source watershed export of element E to rivers ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{ex,E}$ —elemental E export (or “uptake”) from land surface as crop harvest and animal grazing ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{fe,E}$ —elemental E input to watershed (land) from diffuse fertilizer sources ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{fix,ant,N}$ —N input to watershed (land) from agricultural N fixation ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{fix,nat,N}$ —N input to watershed (land) from N fixation on natural areas ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{ma,E}$ —elemental E input to watershed (land) from diffuse animal manure sources ($\text{kg km}^{-2} \text{ year}^{-1}$); $WSdif_{nat,E}$ —net E input to watershed (land) from diffuse sources, on natural areas ($\text{kg km}^{-2} \text{ year}^{-1}$); $WShwDet_P$ —raw elemental P human waste input of detergent to watershed ($\text{kg km}^{-2} \text{ year}^{-1}$); $WShw_E$ —a gross human-waste source to the watershed, where $WShw_N = WShwExc_N$ and $WShw_P = WShwExc_P + WShwDet_P$ ($\text{kg km}^{-2} \text{ year}^{-1}$); $WShwExc_N$ —raw elemental N human waste input of excrement to watershed ($\text{kg km}^{-2} \text{ year}^{-1}$); $WShwExc_P$ —raw elemental P human waste input of excrement to watershed ($\text{kg km}^{-2} \text{ year}^{-1}$); Yld_F —basin yield from nutrient form F, kg N or P per km^2 of basin area per year, simplified as $\text{kg km}^{-2} \text{ year}^{-1}$ ($\text{kg km}^{-2} \text{ year}^{-1}$)

Table 2 List of management strategies to reduce nutrient export to the coastal waters of China and their estimated potential effect on identified model inputs as assumed in the Reduction Scenarios (in % relative to the Base Scenario)

Reduction Scenarios (RSs)	Management strategies	Assumed changes in identified model inputs as used in reduction scenario for period 2000–2050 (in % relative to the Base Scenario)
Fertilizer management (RSF)	<p><i>Match N and P supply with crop demand:</i></p> <p>Test fertilizer N and P needs for different areas, minimize fallow periods to limit mineral N and P accumulation, etc</p> <p><i>Tighten nutrient flow cycles:</i></p> <p>Integrate animal and crop production systems, improve collection and reuse of manure in crop production, substitute N and P fertilizers by the extra available animal manure and human N and P, maintain plant residue on the farmland, etc</p> <p><i>Use advanced fertilization techniques:</i></p> <p>Controlled release fertilizers, deep placement, placing fertilizer in bands or nests, match fertilizer type to seasonal precipitation</p> <p><i>Reduce animal manure excretion:</i></p> <p>Low-N diets for animals, Changing human diets to reduce meat consumption, reducing animal numbers</p> <p><i>Manure regulations:</i></p> <p>Integrate management in animal raising and grazing, manure collection and stock</p>	<ol style="list-style-type: none"> 1. Reduced synthetic fertilizer N and P inputs to watersheds by 40% (including substitute by manure and human excrements); 2. Reduced manure inputs by 20%; 3. Reduced sewage inputs to rivers from human excrement by 50% (We assume that the amount of N and P contained in urine and feces will be used in agricultural production to replace synthetic fertilizers and in the same way as animal manure (Bouwman et al. 2009))
Sewage management (RSS)	<p><i>Reduce sewage emissions to watersheds, reuse human waste and improve sewage treatment technology:</i></p> <p>Recollection and reuse of human excrements in sewage, using P-free detergents to reduce phosphorous emissions from detergents, improve sewage treatment</p>	<ol style="list-style-type: none"> 1. 50% of human excrements is used in agriculture as manure; 2. Reduced P inputs to watersheds from detergents by 20%; 3. Improved N and P removal efficiency in sewage treatment by 20%
Energy management and climate constraints (RSE)	<p><i>End-of-pipe and international air pollution control</i></p> <p>Reduced NO_x and NH₃ emissions, and thus reduced atmospheric N deposition in Chinese watersheds through end-of-pipe technologies, global energy saving and fuel switch,</p> <p>reduced climate impacts on runoff</p>	Reduced atmospheric N deposition and runoff as in the Millennium Assessment Scenarios- TechnoGarden scenario (Bouwman et al. 2009; Fekete et al. 2010): atmospheric N deposition is by 2050 around 50% lower in the selected river basins than in the 2050 GO Scenario
Combined reduction options (RSC)	<p><i>Combined options to reduce inputs from fertilizer, sewage, energy consumption and considering the climate constraints:</i></p> <p>Combination of scenarios RSF, RSS and RSE</p>	<ol style="list-style-type: none"> 1. Reduced synthetic fertilizer N and P inputs to watersheds by 40% (including substitution by manure and human excrements); 2. Reduced animal manure inputs by 20% (partly replaced by human excrements); 3. 50% of collected human excrements is used in agriculture as manure; 4. Reduced sewage inputs to watersheds from detergents by 20%; 5. Improved removal efficiency in sewage treatment by 20%; 6. Reduced atmospheric N deposition and runoff as in the Millennium Assessment Scenario- TechnoGarden scenario (Bouwman et al. 2009; Fekete et al. 2010)

the Three Gorges Projection in the Changjiang. The analyses for the future nutrient export by the selected Chinese rivers for the Global Orchestration scenario are based on the NEWS-2 model outputs.

For the Reduction Scenarios, we assume implementation of the reduction strategies listed in Table 2. We distinguish between four scenarios, assuming management options for agriculture (RSF) and sewage control (RSS). In addition,

energy and climate policies may affect nutrient export by rivers as assumed in scenario RSE. The last scenario combines all three strategies (RSC). Our assumptions for the scenarios are based on the literature and expert judgment and reflect reductions that we consider feasible by the year 2050. Any assumption for the coming decades is surrounded with uncertainties, and the scenarios should therefore be considered indications of possible change rather than predictions of future trends. Nevertheless, they indicate what would happen with the coastal environment if we would be able to reduce N and P inputs as assumed in Table 2.

Data

Our analysis is based on the *NEWS*-2 model runs for the selected 13 Chinese river basins. Basins are defined using STN-30p (version 6.01), a global $0.5^\circ \times 0.5^\circ$ grid dataset (Vorosmarty et al. 2000a, b). For details of the basin attributes creation, we refer to Mayorga et al. (2010). The model input datasets cover climate, hydrology, land use, socioeconomic driving forces, and point and diffuse sources of nutrient inputs. Inputs are spatially distributed at $0.5^\circ \times 0.5^\circ$ resolutions and aggregated to basin scale. The model outputs include river exports and different levels of source contribution. The *NEWS* models calculate river export of nutrients both in terms of yield ($\text{kg km}^{-2}\text{year}^{-1}$) and load (kg year^{-1}) at river mouths.

Nutrient exports at river mouths in the past and present were computed using historical data for the year 1970 and 2000 (Qu and Kroeze 2010). Model inputs for the Base Scenario for the year 2030 and 2050 were derived from the Global Orchestration scenario of MEA. *NEWS* models use several anthropogenic driving forces such as population, GDP and crop-livestock production, from the MEA scenarios directly. Additional nutrient management scenarios and the input datasets were developed by interpreting the MEA storylines (Seitzinger et al. 2010). Inputs from agricultural areas are the surface N and P balances, which were computed by the IMAGE model (version 2.4) and downscaled to basin level (Bouwman et al. 2009). Sewage inputs were estimated from country-specific projections for population and economic growth, urbanization, development of sewage systems and wastewater treatment installations (Van Drecht et al. 2009). Scenarios for *NEWS* hydrological inputs were developed by the Water Balance Model (WBM_{plus}) based on the scenarios information taken from the IMAGE model (Fekete et al. 2010). Inputs for the Reduction Scenarios for future are developed based on the datasets of the Base Scenario and the reduction strategies.

Results and discussion

Export of nutrients by rivers 1970–2000

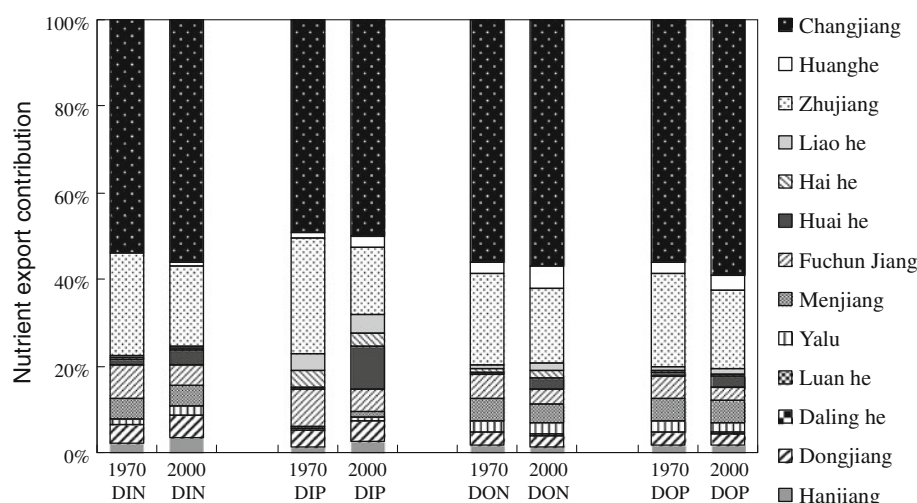
Past trends in nutrient export are important for management decisions and for analyses for future trends. We briefly describe the magnitude of nutrient export and sources, with a focus on the relative contribution of rivers. For details on past trends in nutrient export of Chinese rivers, we refer to earlier analyses (Qu and Kroeze 2010).

From 1970 to 2000, the river export of dissolved forms of N and P from the selected 13 basins increased dramatically, especially for DIN and DIP. DIN export increased more than one fold from 0.7 to 1.6 Tg N year^{-1} . DIP export increased three folds from 0.02 to 0.08 Tg P year^{-1} . Smaller but substantial increases (by about one-third) are calculated for DON and DOP export. DON export increased from 0.23 to 0.31 Tg N year^{-1} . DOP export increased from 0.012 to 0.016 Tg P year^{-1} . The relatively fast increases in dissolved inorganic N and P were also reflected in the load ratio of DIN/ DON (from about 3:1 to 5:1) and DIP: DOP (from 1.5:1 to 4.7:1).

In an earlier study, we compared our modeled nutrient yields to measured values, indicating that the *NEWS* models perform well for dissolved N and P export by large Chinese rivers (Qu and Kroeze 2010). However, there are still considerable uncertainties, in particular for dissolved organic and N and P (DON and DOP). Publications on concentrations of DON and DOP in Chinese rivers are scarce, especially for measurements over a full year. Here, we illustrate the uncertainties by comparing different measured yields for the Changjiang and the Zhujiang. These two rivers contribute by about half and one-fifth of the total nutrient export by the selected 13 river basins in our study, respectively. Liu et al. (2003) estimated the annual yields of dissolved organic N and P in the major tributaries of the Changjiang based on an expedition in April–May 1997. Their measurements indicate that DON and DOP export by Changjiang amounts to about 0.2 Tg N year^{-1} and 0.1 Tg P year^{-1} , respectively. Yu et al. (2009) calculated daily DOP fluxes in the Datong hydrological station in the lower Changjiang, based on measurements in April and September 2006. Based on this, we estimated DOP export to be about 0.01 Tg P year^{-1} . Lu et al. (2009) reported an annual flux of DON from the Zhujiang estuary to the coastal ocean of about 0.1 Tg based on measurements taken from March 2005 to February 2006. Despite the lack of long-term monitoring studies, we consider the quality of our model results to be appropriate for the purpose of our study (Qu and Kroeze 2010).

Nutrients in rivers are from natural and anthropogenic sources. In general, natural sources dominated in the past, but the share of natural sources decreased during the last

Fig. 3 Share of individual rivers in the total nutrient export by the selected 13 rivers in 1970 and 2000. Results are shown for dissolved inorganic N and P (DIN and DIP) and for dissolved organic N and P (DON and DOP). *NEWS-2* model results



30 years, while anthropogenic sources increased. Between 1970 and 2000, the inputs from synthetic fertilizers and sewage increased by a factor of 6 (though sewage is a minor source). Inputs from manure, agricultural N_2 fixation and atmospheric N deposition doubled to tripled. Synthetic fertilizers became the largest single source (responsible for about 40% of DIN in rivers). Other important sources are manure and atmospheric N deposition. The share of natural N_2 fixation in the DIN export decreased from 33% in 1970 to about 6% in 2000. Sewage was the largest single source of DIP in rivers, both in 1970 and in 2000. The share of sewage-DIP in total DIP export increased from one-third in 1970 to two-thirds in 2000. DIP export from synthetic fertilizers and sewage both increased by more than 500%. River DIP from manure increased by about 170%. Weathering became a small source of DIP, from more than 20% in 1970 to only 5% in 2000. River DON and DOP originate largely from leaching. Leaching from natural areas was the individual source (responsible for about two-thirds of DON and DOP) in 1970, while anthropogenic leaching accounted for half of DON and DOP in rivers in 2000. Sewage became a more important source of DON, while synthetic fertilizers and manure became important sources of DOP. The *NEWS* model results indicate that the past increases in nutrient export to the coastal waters of China largely result from increased agricultural production and sewage.

As mentioned in Sect. 2.1, the selected river basins differ with respect to basin areas, population, hydrology and other factors. Seitzinger et al. (2005, 2010) showed that regional trends in nutrient export may differ largely from continental trends. Detailed analyses of individual basins are needed to better understand the spatial variations in nutrient fluxes and to identify appropriate management options.

Figure 3 shows the share of individual rivers in the total nutrient load transported to the coastal waters of China.

The Changjiang alone contributes by about half of the total nutrient export by the selected 13 river basins. Nutrient export by the Changjiang increased fast over time. This is consistent with several other studies, e.g., Li et al. (2007) reported fast increasing trends in DIN and DIP concentrations at the Datong gauging station in the lower Changjiang since 1980s. Yan et al. (2010) described the increasing nitrogen inputs and riverine DIN exports from the Changjiang river basin. The Changjiang has the largest basin area and largest river discharge of the 13 rivers and includes intensive agriculture. About 40% of the total rural and urban population of these 13 river basins live in the Changjiang basin. Figure 4 shows the dominant sources of different forms of nutrients in rivers. For DIN, DON and DOP in the Changjiang, natural sources dominated in 1970 (Fig. 4a, e, g), and anthropogenic sources in 2000 (fertilizer for DIN, see Fig. 4b; anthropogenic leaching for DON and DOP, see Fig. 4f, h). Sewage was the dominant source of DIP in the Changjiang in 1970 and 2000 (Fig. 4c, d).

Another large river is the Zhujiang (Pearl River), the second largest and the third longest river in China. The Zhujiang exports about 20% of the nutrients to the coastal waters of China (more than 20% in 1970, less than 20% in 2000, Fig. 3). Export of dissolved inorganic N and P, especially DIP, by the Zhujiang increased rapidly. DON and DOP export increased slightly. The Zhujiang basin is well-known for its rapid economic growth and urbanization in its delta area (Pearl River Delta, PRD). The estuary of Zhujiang was reported as one of the most polluted coastal waters (SOA 2007). DIN in the Zhujiang stems largely from natural N_2 fixation in 1970 and synthetic fertilizers in 2000 (Fig. 4a, b). The dominant sources of DIP changed from manure (in 1970) to sewage (in 2000) (Fig. 4c, d). For DON and DOP, the dominant sources were natural leaching in 1970 (Fig. 4e, g) and anthropogenic leaching in 2000 (Fig. 4f, h). Both the Changjiang and Zhujiang estuaries

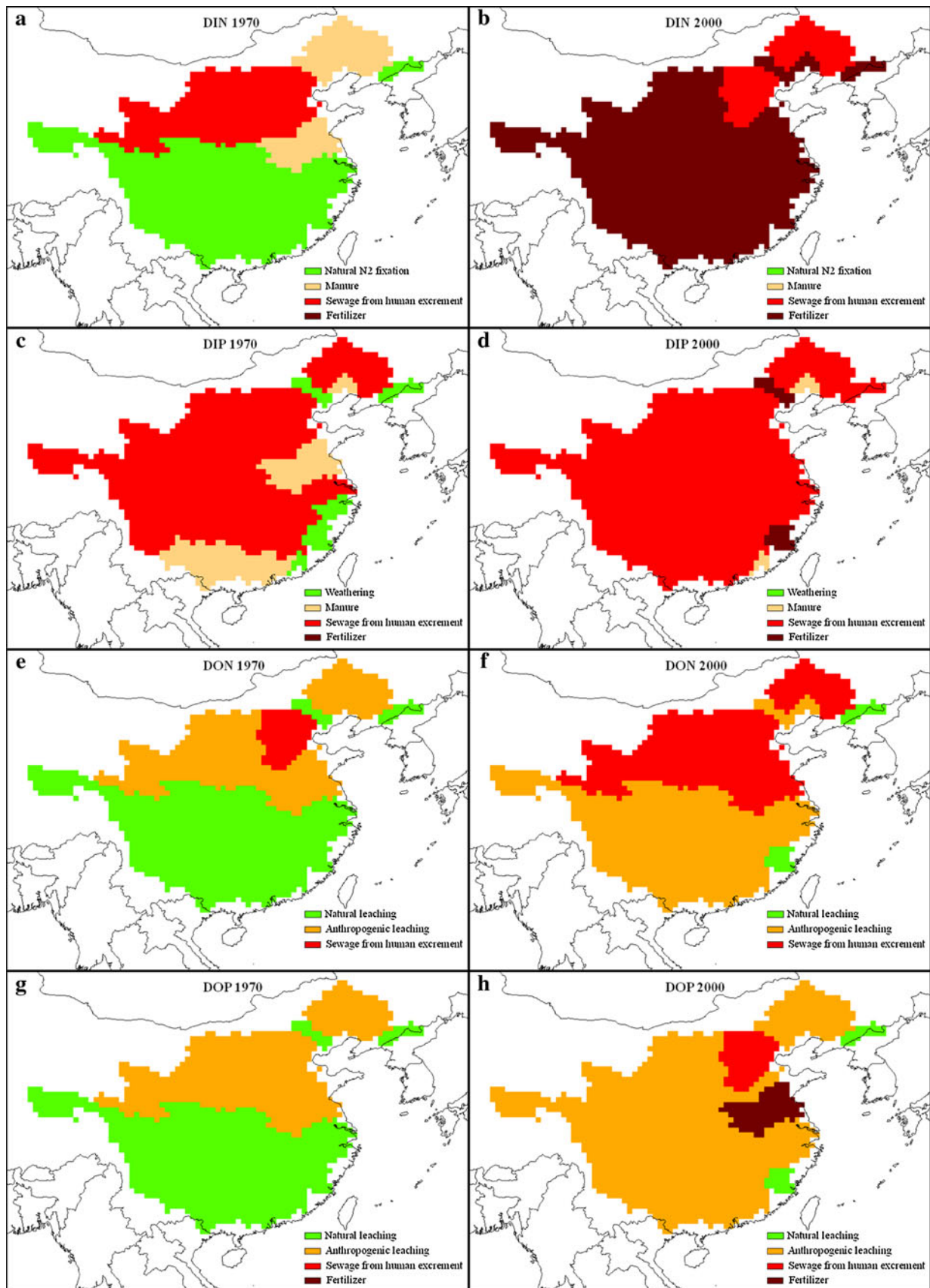


Fig. 4 Dominant sources of nutrients exported to the coastal waters of China in 1970 and 2000. *NEWS-2* model results

were reported as “eutrophication-associated dead zones” (Selman et al. 2008).

Compared with the Changjiang and Zhujiang, the nutrient exports by other rivers are relatively small. The Huanghe (Yellow River) is the second longest river in China, but its nutrient exports to coastal waters are relatively small. This is largely because of consumptive water use. The Huanghe is the most important source of water in northern China and its basin is one of the most important agricultural areas. During the past 40 years, the population in the Huanghe basin has been growing and the economy developed fast. As a result, nutrient inputs to the Huanghe basin increased considerably (Wang et al. 2009). The *NEWS* results, however, indicate that this does not lead to large nutrient export rates to the coastal waters, especially not for DIN. This result is consistent with several other studies. Meng et al. (2007) reported that the nitrogen transport by the Huanghe has been decreasing significantly since the 1990s as a result of damming and excessive consumptive water use. Hydrology is a critical driving force of the nutrients export by rivers (Fekete et al. 2010). The climate of the Huanghe river basin is dry and its annual precipitation is lower than in the other 13 basins. Water discharge by the Huanghe has been decreasing since the 1970s and decreased relatively fast since the 1990s. Extensive water use and extractions are among the important reasons (Chen et al. 2004). The dominant sources of DIN in the Huanghe were human sewage in 1970 and synthetic fertilizers in 2000 (Fig. 4a, b). The dominant sources of DON changed from anthropogenic leaching (in 1970) to human excrement sewage (in 2000) (Fig. 4e, f), while the dominant source of DOP was anthropogenic leaching in 1970 and 2000 (Fig. 4g, h). Sewage was the most important source of DIP in the Huanghe in both 1970 and 2000 (Fig. 4c, d).

The other rivers are smaller. Nevertheless, the Huaihe and rivers located in southern China (Fuchunjiang, Menjiang, Dongjiang and Hanjiang) export more nutrients to coastal waters than the large Huanghe. The Huaihe river is a typical example with fast increasing trends in nutrients export. The annual precipitation of Huaihe is about 900 mm year⁻¹. Many cities and industries are located along this river. The population density and urban population density of the Huaihe are both the highest among the selected 13 basins, which explains the abruptly increase in the DIP export in 1970–2000. The dissolved N export by the Huaihe increased close to five times from 1970 to 2000. In 2000, the DIN and DOP in Huaihe were dominated by synthetic fertilizers, while DIP and DON were dominated by sewage from human waste. The rivers located in northern China, include the Liaohe, Haihe, Luanhe and Daling he, are relatively dry basins (with precipitation of about 500 mm year⁻¹ or less) and have low-nutrient export

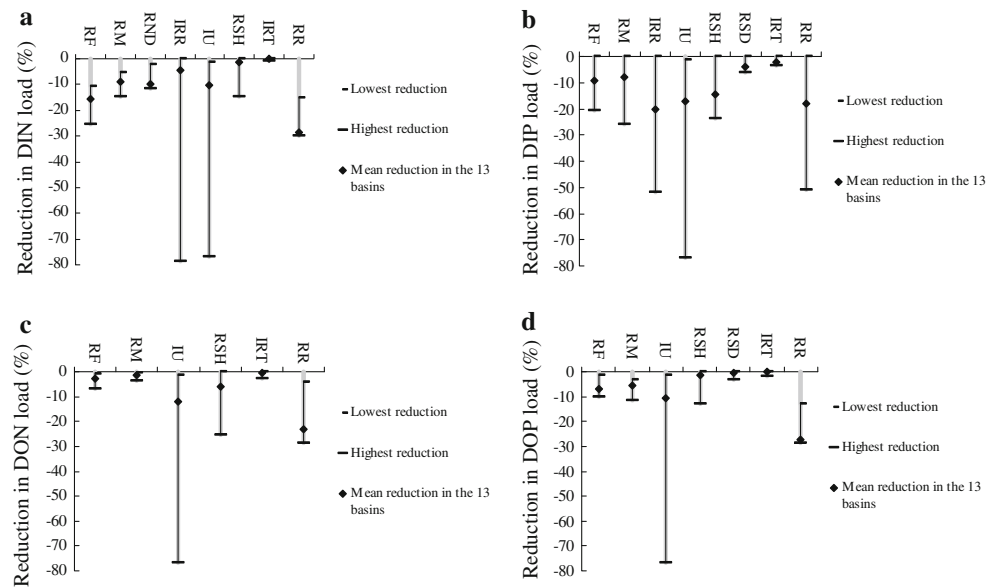
rates. Sewage dominated almost all forms of nutrient export by Haihe (Fig. 4), which can be explained by the high level of urbanization. Many big cities are located in Haihe basin, such as Beijing and Tianjin. The DIP in the Liaohe and the DON and DOP in Yalu also need attention. Nutrient export by Luanhe and Dalinghe was relatively small (or underestimated), because of low population densities, and low river discharge and runoff.

Sensitivity analysis

From the analyses above, we identified several important factors that influence nutrient export to the coastal waters of China. Next, we elaborate the sensitivity of calculated nutrient export by rivers to changes in the identified factors. Figure 5 indicates that the changes in model parameters have different effects on different basins. The ranges in reductions in nutrient export, reflecting the sensitivities of individual basins, are large. This indicates that basin-specific strategies are needed to effectively reduce nutrient inputs to coastal waters. In the following, we will elaborate on this.

Changes in runoff have the largest effect on DIN export (about 30% on average, Fig. 5a). This holds for most rivers, except for the Huanghe, Liaohe and Haihe, which are drier than other basins. Synthetic fertilizer has the second largest impact on DIN export. A 30% reduction in fertilizer inputs reduce DIN export by the selected 13 rivers by 15 (11–26)%. The largest effects of changes in synthetic fertilizer are calculated for the Luanhe, Huaihe, Fuchunjiang, Dongjiang and Dalinghe (>20% reduction), and the smallest for the Zhujiang, Liaohe and Haihe (around 10%). A 30% reduction in manure inputs and atmospheric N deposition both result in around 10% reduction in DIN export by the selected 13 rivers. DIN export decrease by more than 5% for all basins when reducing N and P inputs from manure inputs by 30%. Changes in atmospheric N deposition have the largest effects on the DIN export by the Changjiang and Zhujiang (more than 10% reduction). The smallest reduction is calculated for the Haihe (about 2%). The reduction in DIN export varies significantly among rivers when improving the reservoir retention (from 0 to about 80%). The largest effect is calculated for the Huanghe, which has many large dams and reservoirs in its main stem. A smaller but still considerable reduction is calculated for the Haihe (close to 20%). The Changjiang has a similar response to changes in reservoir retention as the mean of the selected 13 rivers (about 4%). The effect of changes in consumptive water use varies significantly among rivers. The mean reduction for the 13 selected rivers is about 10% and for different basins the changes range from 1% (for the Yalu) to close to 80% (for the Haihe). Changes in sewage inputs from human waste have little

Fig. 5 Results of sensitivity analysis: percentage reduction in nutrient export by the selected 13 basins. Unit: % reduction relative to the year 2000. *RF* reduced synthetic fertilizer inputs to watersheds, *RM* reduced manure inputs to watersheds, *RND* reduced atmospheric N Deposition inputs to watersheds, *RSH* reduced sewage inputs to rivers from human excrement, *RSD* reduced sewage inputs to rivers from detergents, *RR* reduced runoff, *IRR* increased nutrient retention in reservoir, *IU* increased consumptive water use, *IRT* increased nutrient removal by sewage treatment



effects on DIN export for most of the rivers, except for the Haihe, Liaohe and Huanghe (about 14, 10 and 9% reduction, respectively). The reduction in DIN export by any of the 13 selected rivers is less than 1% when improving the sewage removal efficiency.

River export of DIP appears most sensitive to changes in hydrology, including reservoir retention, consumptive water use and runoff (reducing DIP export by the 13 rivers by about 20, 17 and 18%, respectively, Fig. 5b). The response of DIP export to changes in reservoir retention varies among basins. We assume that the reservoir retention for DIP in the Huanghe will not increase any more, arguing that the DIP retention factor in 2000 may have reached its maximum (set by the *NEWS* model), due to the existing large dams and reservoirs in the main stem of the Huanghe. Changes in reservoir retention have large effects on the DIP export by the Zhujiang, Changjiang, Menjiang, Yalu and Haihe (reducing DIP export by 20–50%). DIP export by the selected 13 rivers is found to be sensitive to changes in runoff (reducing DIP export by about 18%), but varies largely among rivers (from less than 1% to about 50% reduction). The largest effects are calculated for the Changjiang, Zhujiang, Menjiang, Hanjiang, Dalinghe and Luanhe (reducing DIP export by 20–50%). DIP export by the three rivers located in northern China (Huanghe, Liaohe, Haihe) is reduced slightly (less than 1%). DIP export by the selected 13 rivers has a similar response to changes in consumptive water use as DIN export.

Dissolved organic leaching is the largest source of DON and DOP in the selected 13 rivers. It should be noted that organic matter leaching from natural and agricultural fields is difficult to quantify. The *NEWS* models use an export coefficient method for this. We, therefore, do not discuss

the sensitivity of the dissolved organic N and P export by rivers to the changes in leaching.

We tested the responses of dissolved organic N and P export to changes in hydrology, including runoff and consumptive water use. The 30% change in runoff reduces dissolved organic N and P export by the selected rivers by more than 25%, except for the Huanghe, Liaohe, Haihe and Huaihe. Changes in consumptive water use have a similar effect on dissolved organic N and P export as on dissolved inorganic N and P export by rivers. Other factors have minor effects. DON export is more sensitive to changes in sewage from human waste than to synthetic fertilizer and manure. DOP export is more sensitive to changes in synthetic fertilizer and manure (Fig. 5c, d).

Based on the sensitivity analyses, we selected a number of promising options to include in our Reduction Scenarios. These include options to reduce nutrient export by Chinese rivers, include reducing fertilizer inputs to agriculture (synthetic fertilizer and manure), reducing sewage emissions to rivers and reducing atmospheric N deposition in the watershed. In addition, the calculated river export of nutrients is found sensitive to changes in water runoff as influenced by climate change in the *NEWS* models. Thus, we include changes in runoff resulted from climate change in our Reduction Scenarios. We do not consider improving nutrient retention in reservoirs and increasing consumptive water promising options, even though this would lead to reduced nutrient transport by rivers. We, however, consider the potential environmental and social tradeoffs too large. In particular, dam construction may impact the environment and ecosystems, as illustrated by the many existing large dams and reservoirs and the associated water withdrawals, especially in the Huanghe basin. Thus, the

alternative scenarios follow the Base Scenario, assuming increased damming of rivers and increased water use in several rivers (see Sect. 5).

Next, we formulated several management strategies and possible actions, and their estimated potential effects on changes in model inputs as used in the Reduction Scenarios (Table 2). In the following, we present results for these scenarios with respect to future trends in river export of nutrients. These scenarios add some useful perspectives to the sensitivity analyses for exploring effective options to control nutrient export to the coastal waters of China in future.

Future trends in nutrient export in the year 2030 and 2050

We first analyze the Base Scenario for some important driving forces of nutrient export by the selected rivers. Details on the four Reduction Scenarios are listed in Table 2.

Population and economy are among the most important driving forces. The population grows slowly in the Base Scenario, while the economy develops fast. The total population in the selected 13 basins peaks at about 1,080 million in 2030 in this scenario, compared with 1,000 million in the year 2000. From 2030 to 2050, the population is projected to decrease by about 70 million. Per capita GDP increases from \$3,550 person⁻¹ year⁻¹ in 2000 to \$15,926 person⁻¹ year⁻¹ in 2030 and \$38,416 person⁻¹ year⁻¹ 2050 (in constant US dollars of 1995).

In the Base Scenario, population growth and economic developments are accompanied with a fast increase in the urban population between 2000 and 2030. In addition, the sanitation is projected to improve quickly. Urbanization slows down between 2030 and 2050, but the fast improvement of sanitation will continue. As a result, the population with access to sewage systems keeps rapidly increasing from 2000 to 2050 (increases of about 170%). The N and P removal efficiencies in waste water treatment systems also increase fast in this scenario. The net effect of these trends is that N and P emissions from human waste to rivers increase dramatically between 2000 and 2030 (increases by about 190% for N and about 170% for P). From 2030, sewage inputs from human wastes increase less fast (for N) or even decrease (for P). The net P inputs from detergents are roughly tripling between 2000 to 2030, and increasing by about one-third from 2030.

The agriculture area in the Base Scenario is assumed to decrease slightly from 2000 to 2050, but the application of N and P fertilizers and inputs from manure increase fast as a result of increased demand for meat and cereals. The N inputs to watersheds from fertilizers increase by about one-fifth in 2000–2030 and 2030–2050. The P inputs to

watersheds from fertilizers increase faster between 2000 and 2030 (by about 170%) than between 2030 and 2050 (by about 20%). Substantial increases are projected for the N and P inputs from manure (increase by about 70% from 2000 to 2030 and 10% from 2030 to 2050).

Hydrological factors are critical driving forces of nutrient export by rivers (Fekete et al. 2010). Future hydrology may be different because of climate change, damming of rivers and human water consumption. In the Base Scenario, the river discharge and runoff by the Changjiang and the rivers in southern China are projected to increase between 2000 and 2050 as a result of climate change. For the Huanghe and the rivers in northern China, the trends are different for 2000–2030 and 2030–2050. Nutrient retention by damming and in reservoirs is projected to increase as a result of hydropower development. In addition, consumptive water use keeps increasing between 2000 and 2050.

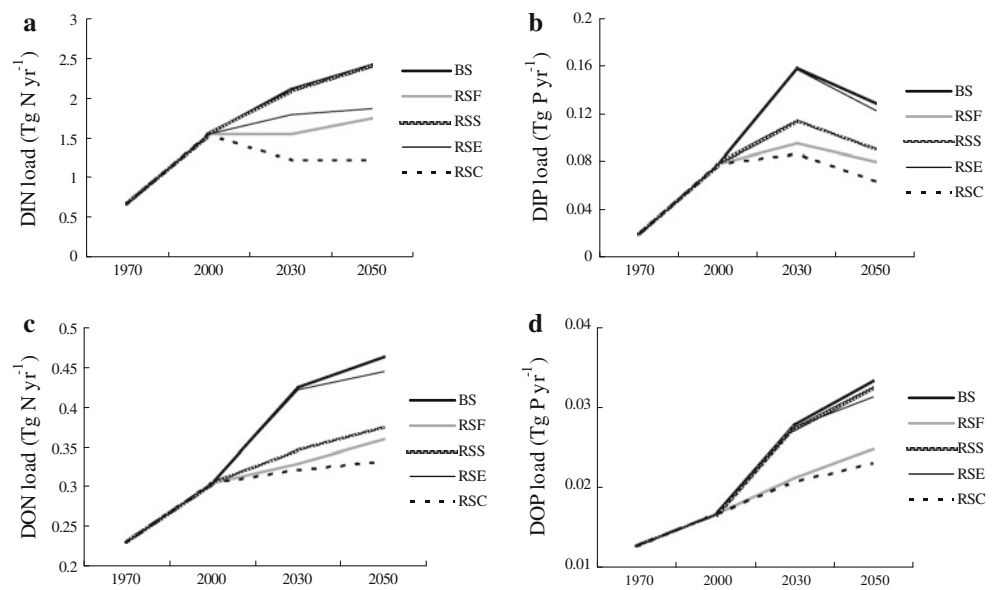
Figure 6 shows future trends in nutrient export in the years 2030 and 2050 for the Base Scenario and the four Reduction Scenarios.

For the Base Scenario, we calculate that dissolved N and P export to the coastal waters by the selected 13 rivers increase considerably between 2000 and 2030. This is especially the case for dissolved P export. DIP export is calculated to double, while DOP export increases by two-thirds. Dissolved N export increases by less than 40% (37 and 39% for DIN and DON, respectively) (Fig. 6). The increasing trends in river export of nutrients in the Base Scenario continue between 2030 and 2050 except for DIP (14, 9 and 19% increases for DIN, DON and DOP, respectively, and 19% decrease for DIP).

For the Reduction Scenarios, we calculate considerably lower nutrient export rates than for the Base Scenario. This indicates that the management strategies in the Reduction Scenarios may reduce N and P inputs to coastal waters of China. River export of DIN is affected mostly by reduction strategies for agriculture and energy use: DIN loads for the selected 13 rivers are 27–28% lower in the RSF scenario than in the Base Scenario (Fig. 6a). In the RSE scenario, DIN export is projected to be 15% lower in 2030 and 23% lower in 2050 than in the Base Scenario. Policies focusing on sewage control may be less effective for DIN: in the RSS scenario, DIN export by rivers is hardly lower than in the Base Scenario (about 1% in both 2030 and 2050). When we combine all the strategies in the RSC scenario, DIN export is reduced by 42% in 2030 and 50% in 2050 compared with the Base Scenario.

River export of DIP is affected most by changes in agriculture and sewage: the calculated DIP export by rivers is 40 and 30% lower in RSF and RSS scenarios than in the Base Scenario, respectively (Fig. 6b). Energy policies are less effective: in the RSE scenario, the DIP export is a few

Fig. 6 Nutrient export to the coastal waters of China by the selected 13 rivers for the Base Scenario (BS) and for the Reduction Scenarios for the period 1970–2050. See Table 2 for an overview of the scenarios



percent lower in 2030 and 2050 compared with the Base Scenario. The combined strategies in the RSC scenario result in a 50% reduction in DIP export in 2030 and 2050. For DON, we also calculate reductions in the RSF and RSS scenarios, reducing DON export by about 20% relative to the Base Scenario (Fig. 6c). The RSE scenario has little effect on DON and DOP export. The combined strategies in the RSC scenario reduce DON export by up to 28% in 2050. The reduction in DOP export is significant in the RSF scenario (about 25%), but minor in other scenarios (a few percent) (Fig. 6d). In the RSC scenario, DOP export is projected to decrease by up to 30% compared with the Base Scenario.

From above analyses, we may conclude that the combined strategies may have significant effects on reducing continental river export of all forms of dissolved N and P to the coastal waters of China. In this scenario, river export of dissolved inorganic N and P is about 40–50% lower than in the Base Scenario and of dissolved organic N and P about 20–30% lower. The effectiveness of reduction options differs between strategies and nutrient forms. In general, changes in agricultural practices have the largest effect on future N and P river export. Strategies for sewage are most effective for reducing DIP and DOP export by rivers. Climate and energy policies have a considerable effect on DIN river export.

Future trends in nutrient export differ among rivers and scenarios. In the following, we present results for the Changjiang, Huanghe and Zhujiang in 2030 for three Reduction Scenarios (RSF, RSS and RSE) as illustrative examples. These are the largest rivers in China in terms of length or water discharge. As the most important agricultural areas and economic zones of China, these basins support half of the country's population.

We calculated relatively large reductions in DIN export by the Huanghe for the RSF scenario (about 40%) and the RSS scenario (22%). For the Changjiang and Zhujiang, the RSF and RSE scenario are relatively effective, reducing DIN export by 21–27% relative to the Base Scenario. The largest reductions in DIP export by the Changjiang and Zhujiang are calculated for the RSF scenario (by about 40%) and by the Huanghe for the RSS scenario (by close to 50%). In the RSE scenario, DIP reductions are relatively large for the Changjiang (6%) and Zhujiang (7%). For DON export, the largest reductions are calculated for the Huanghe (by about 40% in RSF and RSS scenarios). The RSF and RSS scenarios are also effective on reducing DON export by the Changjiang and Zhujiang (9–20%). For DOP export, large reductions are calculated for the Huanghe for the RSF scenario (by 28%) and the RSS scenario (by 16%), for the Changjiang and Zhujiang for the RSF scenario (25% for the Changjiang, 21% for the Zhujiang). The RSS scenario is not effective for reducing DOP export by the Changjiang and Zhujiang. In the RSE scenario, reductions in DON and DOP export are relatively large for the Changjiang and Zhujiang (by 5–6%).

Trends in future nutrient export by individual rivers differ among scenarios. Here, we present the nutrient export by the largest river (the Changjiang) for three Reduction Scenarios (RSF, RSS and RSE). DIN export by the Changjiang is significantly lower in the RSF and the RSE scenarios than in the Base Scenario (27 and 24% lower, respectively). The difference in DIN export between the Base Scenario and RSS scenario is minor. DIP export by the Changjiang is 40% lower in the RSF scenario than in the Base Scenario. For the RSS and RSE scenarios, we calculate the DIP export to be 23 and 6% lower, respectively. DON export is 20% lower in the RSF than in the

Base Scenario. Reductions in DON export in the RSS and RSE scenarios are 16% and 5%, respectively, compared with the Base Scenario. DOP export is 25% lower in the RSF than in the Base Scenario, while 2% lower in the RSS scenario and 6% lower in the RSE scenario, respectively.

Conclusion

We analyzed past and future trends in nutrient export from the Chinese continent to coastal waters and evaluate the effectiveness of different options to reduce nutrient inputs to the coastal waters of China.

Between 1970 and 2000, river export of dissolved N and P increased significantly. River export of DIN doubled and of DIP tripled over this period. The increase in dissolved organic N and P was more moderate (about 30%). Over time, anthropogenic sources of N and P in river water became more important. In particular, N and P losses from agriculture increased. Currently, more than half of the dissolved N and P in Chinese rivers are from agriculture. In 1970, this was about one-third. The relative shares of different N and P sources differ among basins. In some basins, agricultural sources dominate, while in others sewage is the dominant source of nutrients in rivers.

The total dissolved N and P export by the 13 rivers analyzed here is dominated by the Changjiang (Yangtze River) and Zhujiang (Pearl River). About 50% of the total dissolved N and P inputs by the 13 rivers to the Chinese seas are from the Changjiang, and about 20% from the Zhujiang.

We analyzed the sensitivity of modeled N and P export by rivers to changes in model inputs and parameters. In this sensitivity analysis, we focused in particular on model inputs and parameters reflecting human influences. The modeled export rates appear particularly sensitive to changes in hydrology, caused by consumptive water use, damming of rivers and runoff (reflecting climate change). In addition, changes in fertilizer use and manure excretion affect the calculated export of dissolved N and P considerably. Changes in sewage inputs have a relatively large effect on DIP export by rivers, and to some extent on DON export.

Next, we analyzed future trends in river N and P export for different scenarios in 2030 and 2050. In our Base Scenario (Global Orchestration scenario), the N and P export by the selected 13 rivers increases considerably (40–100%) between 2000 and 2030. From 2030, the increase continues except for DIP. We formulated four alternative scenarios, reflecting different strategies to reduce N and P export by rivers. These scenarios focus on agricultural management, sewage management and climate and energy policy. We conclude that all three strategies

may reduce the N and P inputs to coastal waters considerably. When we combine the three strategies, the N and P inputs to coastal waters may decrease compared with 2000, more or less stabilize at the 2000 level or increase moderately. Agricultural management has the largest impact on future dissolved N and P river export. Sewage management may reduce the DIP and DON export in particular. The effects of climate and energy policies on dissolved N and P export by rivers are moderate; only for DIN, we calculate a considerable reduction.

Our main conclusion is that without pollution control the export of dissolved N and P by rivers to the coastal waters of China may increase considerably. This increase could be avoided by a mix of management options for agriculture, sewage and energy use. Our second conclusion is that the preferred mix of options differs among basins, in line with differences in sources of N and P and hydrological factors.

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